Governor Frank Keating's
Tar Creek Superfund Task Force

Water Quality Subcommittee
Task 2 Report

Water Quality Improvement Alternatives

September 7, 2000

Task 2 as charged to the Water Quality Subcommittee:

Compile available research on treatment alternatives that would remove the mining-related contaminants that impair the beneficial uses assigned to the surface and ground waters in the former Mining District. The subcommittee shall evaluate the feasibility of constructed wetlands, diversion techniques, drinking water treatment systems, methods of lowering the water table, and any other alternatives identified by Task Force or subcommittee members. In its evaluation, the subcommittee should review the work conducted by the Oklahoma Water Resources Board as part of the task force created during Governor Nigh's administration, as well as any other attempts to address water quality concerns over the past several decades. The subcommittee shall draft a project proposal for each alternative that includes a scope of work, timeline, resource needs (both capital and personnel), and potential sources of funding for the project. Each proposal should also estimate the degree of water quality improvement expected, as well as specific measures for evaluating project effectiveness.

Executive Summary

In accomplishing Task 2 the Water Quality Subcommittee compiled available research on mine drainage treatment alternatives, evaluated the efficacy and applicability of these alternatives and selected preferred alternatives. Specific project proposals cannot be offered at the present time because of the lack of necessary and current water quality and quantity data and the limited time available. Generic project descriptions and rough cost estimates are provided and can be refined, as sufficient data become available.

The Subcommittee considered several mine drainage mitigation measures (i.e., efforts that would decrease or eliminate mine drainage flows). However, due to the vast spatial extent of the water quality problems in the Tar Creek watershed, temporally variable nature of groundwater discharges, lack of sufficient understanding of the subsurface hydrology, and knowledge gained from past failures of similar efforts, the Subcommittee instead focused their efforts on remediation of existing mine drainage discharges (i.e., treatment of poor quality water). Therefore, surface water diversions, water table lowering and similar measures were not considered to be viable. Likewise, due to the apparent success of current efforts to protect the Roubidoux aquifer from further contamination, drinking water treatment was not examined as a necessary alternative.

The Subcommittee considered active and passive treatment alternatives from a process-based approach. Mine drainage treatment technologies rely on any of the following processes, singularly or in combination depending on water quality and quantity: 1) chemical neutralization, 2) aeration, 3) biogeochemical processes and 4) novel active mechanisms, e.g., reverse osmosis, ion exchange resins, or electrodialysis. Due to the near-neutral conditions of recently analyzed mine drainage discharges in the Tar, Lytle and Beaver Creek watersheds (that must be confirmed with additional sampling), chemical neutralization was considered unnecessary and detailed analysis of this process was not conducted. Efforts therefore focused on aeration, biogeochemical processes and novel active mechanisms. All novel active mechanisms were eliminated from further consideration due to their unproven nature and excessive capital, operation and maintenance costs.

Therefore, aeration and a specific biogeochemical process (i.e., bacterial sulfate reduction) were selected as preferred treatment alternatives. Both of these processes may be promoted in ecological-engineered passive treatment systems (i.e., treatment wetlands), which require limited operation and maintenance. However, these processes must be linked in concert to effectively ameliorate water quality degradation in the Tar Creek watershed. Mine drainage discharges in this watershed contain several metal pollutants, some of which are amenable to aeration treatment (iron), while others are more suited to sequestration via sulfate reduction processes (lead, zinc and cadmium). Successful implementation of these technologies may involve minor modification to existing wetland areas on mined lands to increase water residence times and/or construction of *de novo* wetlands at suitable sites. It is anticipated by the Subcommittee that properly designed and sized full-scale passive treatment systems could effectively improve the quality of mine drainage discharges at the Tar Creek Superfund site at a estimated cost of roughly $25 million in initial and capital costs and approximately $5-10K in yearly costs.

In order to accomplish the water quality improvement goal, the Subcommittee provides the following recommendations with estimated start dates and costs.

1) **Comprehensive monitoring of mine drainage discharges (Fall 2000, $1M)**
2) Evaluation of water quality concerns from chat pile and mill-pond runoff (Fall 2000, $1M)
3) Implementation of pilot-scale demonstration systems (Fall 2001, $5M)
4) Coordination with other subcommittees and the U.S. Army Corps of Engineers Wetland Concept Plan (Fall 2000, $100K)
5) Full-scale treatment system implementation (Fall 2005, $18M)

I. Historical Mine Drainage Mitigation at the Tar Creek Superfund Site

A. Summary

In the feasibility study for operable unit (OU1), Surface and Ground Water, (EPA Record of Decision R06-84/004), six mine drainage remediation alternatives and two mitigation alternatives were considered. The remediation alternatives considered were:

a) No action
b) *In-situ* treatment of mine water
c) Pumping and treatment of mine water
d) Treatment of mine water discharges
e) Plugging of abandoned Roubidoux wells
f) Surface water diversions

The mitigation alternatives considered were:

a) Treatment of Roubidoux water supplies
b) Provision of alternative drinking water supplies.

The remedial options implemented under OU1 were 1) plugging of abandoned Roubidoux wells and 2) surface water diversions. All other options considered for remediation and/or mitigation were not implemented. For the no action alternative, it was determined that no action would result in continued environmental damage to Tar Creek and that no action could eventually lead to contamination of the Roubidoux aquifer. In evaluating the potential treatment of Roubidoux water supplies, it was determined that water quality was sufficient so that no treatment was necessary. For the other remediation/mitigation options, it was determined that costs were simply excessive.

Since the issuance of the OU1 ROD, no other remedial or mitigation options of Tar Creek have been formally considered or reviewed. A brief description of all options considered follows.

B. Past investigation and implementation of mitigation measures

1. Objectives and selection of options

The objectives of initial water quality remediation/mitigation efforts at the Tar Creek site were to alleviate the potential threat to public health and the environment by preventing contamination of the Roubidoux aquifer and by minimizing mine drainage releases to Tar Creek. The National Contingency Plan (40CFR-Part 300; 47 FR 31180) states that "the appropriate extent of remedy shall be determined by the lead agency's selection of the remedial alternative which the agency determines is cost-effective (i.e., the lowest cost alternative that is technologically feasible and reliable and which effectively mitigates and minimizes damage to and provides adequate protection of public health, welfare, or the environment)." Two of the eight alternatives (*in situ* treatment and pumping and treatment) initially selected for evaluation addressed cleanup objectives and would most likely accelerate the improvement of ground water. However, both alternatives were eliminated from detailed analysis because they were excessively expensive. The other six alternatives were selected for further assessment and were evaluated on the basis of effectiveness, durability, reliability, ability to be implemented and cost. A cost–effectiveness evaluation was performed on surface and groundwater alternatives. Using this methodology, alternatives were evaluated against each other according to several measures of effectiveness (technology status, risk and effect of failure, level of cleanup/isolation achievable, ability to minimize community impacts during implementation, ability to meet relevant public health and environmental criteria, and time required to achieve cleanup/isolation) and cost.

In addition to the two remedial options implemented, a limited surface and ground water monitoring program was also conducted to maintain the effectiveness of the remedial actions. As reported in the Five Year Review (EPA 1994) the remedial actions considered at the Tar Creek Superfund Site consisted of the following activities.

2. Rejected options

The rejected alternatives consisted of four remediation options (no action, in situ treatment, pumping and treatment and discharge treatment) and two mitigation options (treatment of Roubidoux water supplies and provision of alternative drinking water supplies.)

No action. The no action alternative was evaluated for the purpose of assessing the potential for the system to recover under natural conditions with no outside influence. Studies indicated that it would take 60-100 years to flush the mines of contaminated water (estimated to be 76,000 acre-feet). Using Darcy's equation to calculate the time necessary for mine water to traverse the vertical distance form the Boone to the Roubidoux indicated that approximately 15,000 to 25,000 years would be required. However, studies indicated that abandoned Roubidoux wells were potentially major pathways allowing significant quantities of mine water to contaminate the Roubidoux in a relatively short period of time. The no action alternative was rejected based on the conclusion that no action would result in continued environmental damage to Tar Creek and would allow contamination of the Roubidoux Aquifer.

In situ treatment. *In-situ* treatment consists of treating the source of contamination, i.e., the water in the mines. The *in-situ* treatment technology evaluated consisted of pumping mine water to the surface and slurrying with alkaline materials. The slurried alkaline material would be pressure pumped back into the flooded mine shafts at such intervals and volumes to provide maximum mixing and treatment of the contaminated mine water. Introducing the alkaline slurry materials would result in the elevation of pH and cause the dissolved metals to precipitate as insoluble metal hydroxides. The major disadvantages of *in situ* treatment were the high cost of the enormous quantities of neutralizing materials required due to low pH of the mine water; technical concerns about achieving adequate mixing of the neutralizing agents and acid mine water; and concerns about mine collapse and subsidence from pumping large volumes of water. Based on cost and potential technical problems it was concluded that treating such a large volume (26 billion gallons) of mine water was not feasible.

Pumping and treatment. As described in the ROD, pumping and treating acid mine water consists of the following components: plugging/sealing all known point discharges, surface diversion to reduce periodic inflows of surface water into the underground workings, installation of a collection well system, and construction of a chemical treatment plant to precipitate the heavy metals from the mine water. This alternative was designed to treat all the acid mine water in a relatively short period of time compared to the time required for natural restoration. The major disadvantages that resulted in rejection of this alternative were high capital and operation and maintenance costs, over-pumping of mine waters that could result in mine subsidence, and the production of massive quantities of sludge, which may be a RCRA characteristic hazardous waste.

Treatment of mine discharges. The treatment of mine discharges alternative consisted of collecting and pumping acid mine discharges to a centralized treatment plant (or plants) for treatment and discharge of treated effluent to surface waters. Chemical precipitation of the heavy metals was the preferred treatment technology. No water was expected to flow from the major outflow areas after completion of the diversion work, thereby rendering this option unnecessary. The projected cost for this alternative was approximately twice the cost of the selected alternatives. The major disadvantages that resulted in rejection of this alternative were that only the larger point discharges would be practical to collect and treat (collecting smaller and more diffuse springs, and other outflows would be impractical); high capital, operation and maintenance costs (approximately $30 million) and failure to restore the aquifer to drinking water quality.

Treatment of Roubidoux water supplies. This alternative, to be implemented if the Roubidoux became contaminated, consisted of treating the Roubidoux water supplies by lime softening to remove heavy metals prior to distribution for consumption. The major disadvantages of this alternative were high capital, operation, and maintenance costs; and failure to restore the aquifer to drinking water quality. However, in the event that the Roubidoux aquifer became contaminated, treatment of water produced from Roubidoux wells prior to distribution was considered to be a feasible alternative.

Provision of alternative drinking water supplies. In the event that widespread contamination of the Roubidoux Aquifer was to occur, this alternative consisted of provision of an alternative drinking water supply. Pumping water from Grand Lake to Commerce, Oklahoma was evaluated. After standard treatment, water would be distributed for consumption. The ground water monitoring plan was expected to detect contamination before it became a significant problem. Since the Roubidoux was not contaminated except for a few localized spots, alternative drinking water supplies were not necessary. The cost for building and maintaining this system was estimated to be $17 million. The disadvantages of this alternative were high capital, operation, and maintenance cost; and failure to restore the Roubidoux aquifer to drinking water quality. Additional supply alternatives would need to be considered for the other towns and rural areas if widespread contamination of the Roubidoux were to occur.

3. Selected options

The remedial options implemented under OU1 were 1) plugging of abandoned Roubidoux wells and 2) surface water diversions.

Plugging abandoned Roubidoux wells. Well plugging at the site consisted of clearing the well holes of obstructions and setting an acid resistant cement plug from bottom to top. The well plugging program did not completely mitigate all threats to the Roubidoux aquifer. There are several ways that the polluted Boone aquifer may contaminate the Roubidoux including fractures, unknown abandoned wells and natural flow. If additional abandoned Roubidoux wells were located, additional funds would be required in order to plug them. Therefore implementation of a monitoring program was recommended to detect trends in water quality of the Roubidoux. For the 83 abandoned Roubidoux wells in Kansas and Oklahoma, the average cost of construction per well varied depending upon the difficulty in clearing each well. The actual cost of the well plugging, including remedial design costs and state matching funds, was $2,698,711. This represents an average cost per well of approximately $32,515.

<u>Surface water diversions.</u> Surface water diversion and diking structures were constructed to prevent surface drainage into mineshafts, subsidence areas, and open boreholes. The diversion program constituted rerouting surface flows away from mineshafts, subsidence areas, and open boreholes. Approximately 600 mineshafts and collapse depressions were identified in the study area, each providing avenues for inflow of surface water into the mines. Once water enters the mines, it mineralizes and flows out of springs and boreholes into Tar Creek further downstream. The surface water diversion action targeted three major inflow areas identified as the Muncie, Big John, and Admiralty mines that were estimated to represent approximately 75 percent of the yearly surface inflows into the mine workings. The Admiralty site was an outflow point but it was predicted that after the water level in the mines was lowered (as a result of the mitigation effort) it would become a major inflow point. It was projected that reducing the surface water inflow into the mines by 75 percent would eliminate or decrease by a significant amount the surface discharges of mine water, and also cause the ground water levels in the mines to drop by a significant amount. No numerical target cleanup goals, reductions in mine discharges, or reductions in the ground water level in the mines were stated in the ROD. The actual cost of the surface diversion and diking program, including remedial design costs and state matching funds, was $ 1,576,531.

4. <u>Monitoring of Surface Water and Ground Water</u>

Because the diversion work did not completely stop all surface discharge of mine water, a two year monitoring and surveillance program (1987-1988) assessed the effectiveness of the remedial actions in mitigating contamination of Tar Creek and preventing degradation of the Roubidoux Aquifer. For surface water, flow measurements were made and water quality data were collected to determine if the pollutant loading to Tar Creek was reduced after construction of the diversion and diking structures. Also, water levels in the Blue Goose Mine were monitored, which are considered indicative of the potentiometric surface of the Boone Aquifer, and thus indicative of discharge volumes of mine water into Tar Creek. If there continued to be significant discharge, remedial measures would need to be evaluated to determine if further action was appropriate. For the Roubidoux Aquifer, water quality data were collected from public water supply wells to assess water quality following the well plugging activities. Details of the monitoring program are presented in the “Tar Creek After Action Monitoring Report” (OWRB, 1991). The report concluded that:

a) Concentrations of most constituents in the mine water discharges were decreasing. Although it was not possible to identify the cause of this decrease, it was determined likely that the decrease is a naturally occurring phenomenon.

b) The volume of the mine water discharged to Tar Creek was not significantly impacted by the remedial action.
c) Surface water quality was not significantly improved, and the diking and diversion remedial action was at best only partially affective.
d) Although some public water supply wells in the Roubidoux aquifer are impacted by mine water, insufficient data exist to evaluate the effectiveness of the well plugging operations.

In 1991 development of a second ground water monitoring program was begun. The program was designed to assess the status of the Roubidoux Aquifer and to determine whether the well plugging operations had succeeded in preventing its contamination. A two phased approach was developed which began with wellhead monitoring and concluded with discrete sampling of the Roubidoux Aquifer.

The well head monitoring was conducted in 1991 and 1992. It consisted of monthly sampling for six months of 21 public water supply wells inside and outside of the mining area. The second phase is ongoing and consists of discrete sampling of five impacted wells within the mining area and the installation of five monitor wells constructed to meet public water supply well design standards.

II. Criteria for selection of mine drainage mitigation efforts at Tar Creek

Treatment options for mine drainage may be generically divided into two general types: passive or active technologies. The Spring 2000 report submitted to the Task Force by The University of Oklahoma environmental capstone students provides a detailed description of mine drainage treatment options (University of Oklahoma, 2000).

Active treatment systems involve the addition of alkaline chemicals, mechanical aeration and other constant manipulation. Passive treatment systems, i.e., those that rely on natural biogeochemical and microbiological processes to ameliorate mine drainage problems, provide a viable treatment alternative. Passive systems require less operational and maintenance labor and have lower initial costs but require larger land areas than traditional active chemical treatment systems. The most common passive treatment systems are aerobic wetlands, anoxic limestone drains, organic substrate (e.g., compost) wetlands and successive alkalinity producing systems (SAPS). These low-maintenance and relatively inexpensive natural systems are the only viable option for abandoned mine drainage treatment (Nairn, et al., 1999, 2000; Hedin et al., 1994; Nairn and Hedin, 1992). For selection of either passive and active treatment alternatives, the subcommittee evaluated the following parameters.

A. Target effluent values

Target effluent values must be established prior to implementation of a mine drainage remediation or mitigation technology. Possible criteria include the State of Oklahoma Water Quality Standards, U.S. EPA National Recommended Water Quality Criteria, Safe Drinking Water Act standards or data on nearby reference streams. The quality of mine drainage discharges at the Tar Creek site do no meet any applicable standard.

B. Climatic variations and impacts

Regional climate patterns and their effects on hydrologic variability will influence the selection of mine drainage remediation or mitigation technologies. Biologically based technologies (i.e., treatment wetlands) demonstrate seasonal variability that influences design and sizing considerations. These systems must be designed for year-round effectiveness in meeting target effluent values. In addition, mine discharge flow rates in the Tar Creek watershed demonstrate great variations (over several orders of magnitude) based on regional precipitation and recharge patterns. Therefore, mitigation technologies must take into account variations in mine discharge quantities and qualities and should be designed to effectively mitigate the worse case scenario.

C. Facilities requirements

Any mine drainage mitigation technology implemented will require the acquisition and utilization of land within the watershed. These requirements may vary substantially for specific options. In general, passive treatment technologies require larger land areas (but lower capital and operation and maintenance costs) than active treatment technologies.

D. Management requirements

Mine drainage discharges in the Tar Creek watershed are likely to continue for many decades, thus requiring treatment for an equivalent period of time. Therefore, any mitigation technology to be implemented must require limited operation and maintenance to be feasible in the long term. It is likely that operation and maintenance endeavors will need to be carried out by onsite personnel.

E. Capital costs

Given the scope of the Tar Creek water quality problem, it is anticipated that capital costs for any and all mitigation technologies are likely to be substantial. However, capital costs must be kept within available resources.

F. Operational expenses

Long term (i.e., several decades) of operation and maintenance will likely be required for any selected mitigation measure. Therefore, operating expenses must be sustained for an equivalent length of time.

III. Evaluation of selected technologies

A. Possible technologies considered

Degraded surface and ground water quality is a major environmental concern at many abandoned mining operations and operational experience elsewhere provides a framework for selection of remediation options at Tar Creek. Mine drainage is usually characterized by elevated concentrations of metals (often iron, manganese, aluminum, lead, zinc, cadmium and nickel), acidity and sulfate. Effective treatment of contaminated mine drainage requires remediation of solution pH, Eh and ionic constituency. Consequently, technologies that focus on only one of these parameters cannot effectively treat the Tar Creek mine drainage. Mine drainage treatment technologies may rely on any of the following processes, singularly or in combination depending on water quality: 1) chemical neutralization, 2) aeration, 3) biogeochemical processes 4) other active mechanisms, e.g., reverse osmosis, ion exchange resins, electrodialysis.

Traditional acid mine drainage treatment technologies rely on active additions of highly alkaline chemicals (e.g., sodium hydroxide, calcium hydroxide, etc.) and electromechanical aeration to facilitate metal oxidation, hydrolysis and precipitation under controlled pH conditions. These relatively laborious and cost-intensive operations require regular maintenance and active manipulation and, therefore, are not viable options for most abandoned mines.

It is important to note that waters at the Tar Creek site are contaminated with elevated concentrations of iron, lead, zinc, cadmium and sulfate, but have near neutral pH and contain substantial quantities of bicarbonate alkalinity. Because the shallow Boone aquifer at the Tar Creek site is a cherty dolomite (i.e., $CaMg(CO_3)_2$), subsurface waters have pH values near neutrality (pH 5.5-6.5). The near-neutral pH condition of water discharging to the surface is beneficial to effective mine drainage treatment because a neutralization step has already been accomplished. Consequently, those mine drainage treatment technologies that involve addition of alkaline chemicals in either solid or solution form are not relevant to the Tar Creek site. These somewhat unique water quality characteristics are extremely important for the design and implementation of effective treatment technologies.

Although mine drainage at Tar Creek is near neutral pH, it remains heavily contaminated with iron, zinc, lead and cadmium. As a consequence, effective treatment technologies must be able to decrease contamination of these metals to levels compatible with the designated use of receiving stream waters. However, many technologies focus on neutralization of the acidic nature of mine drainage (see Appendix 1) and are not applicable at Tar Creek. They were therefore eliminated from further evaluation.

Technologies that are not strictly focused on neutralization of mine drainage are listed, briefly discussed, and evaluated in Table 1. Of the five potential technologies, reverse osmosis, ion exchange and electrodialysis are unlikely candidates for any form of implementation at the Tar Creek Site due to their excessive costs. Further, it is clear that the deployment of aeration and sulfate reduction technologies (a biogeochemical process) alone cannot effectively treat mine drainage. This is especially true at Tar Creek because of the high toxic metal burden in the waters.

B. Preferred alternatives selected

At the Tar Creek site, the preferred mine drainage treatment option will link aeration with sulfate reduction in passive treatment wetland systems. In this way the negative ecological effects due to high iron content will be eliminated by aeration, and the deleterious environmental and human health concerns of toxic metals (i.e., lead, zinc and cadmium) in surface waters will be adequately addressed via sulfate reduction.

The dominant treatment processes in aerobic passive treatment systems are iron oxidation, hydrolysis, precipitation and settling. The hydrolysis reaction produces proton acidity, and therefore, this technique is applicable only to net alkaline mine drainages. For the Tar Creek waters, limited aerobic removal of other metals (lead, zinc and cadmium) may occur due to carbonate precipitation upon the substantial degassing of carbon dioxide from the subsurface waters when they are exposed to the atmosphere. Therefore, systems encouraging anaerobic processes, i.e., bacterial-sulfate reduction, must also be utilized to effectively sequester these metals as insoluble metal sulfides in the substrate of the passive treatment system.

The current state-of-the-art in passive treatment technology couples aeration ponds with vertical flow-wetlands that promote bacterial sulfate reduction (e.g., Kepler and McCleary 1994, Nairn et al. 1999; Nairn et al. 2000, Watzlaf et al. 2000; Peart and Cooper 2000; Jage et al. 2000). The dominant physicochemical treatment processes in the aerobic wetlands portion is iron oxidation, hydrolysis, precipitation and settling. In the vertical-flow wetland portion, significant quantities of metals may be sequestered as insoluble sulfides. These systems, sometimes known as successive alkalinity producing systems (SAPS) or reducing and alkalinity producing systems (RAPS) have been implemented at several abandoned mine sites (e.g., Hedin and Nairn 1993, Willow et al. 1998, Crisp et al. 1998, Nairn et al. 1999; Nairn et al. 2000). In addition to these processes, these systems promote the removal of zinc carbonate in a high environment of elevated partial pressures of carbon dioxide (Nuttall 1999, Nuttall and Younger 2000).

The Subcommittee recommends ecological-engineered treatment wetlands as the preferred water quality improvement mechanism at the Tar Creek Site. However, critical data are currently lacking in order to ensure successful implementation of these technologies.

IV. Recommendations

The Subcommittee provides the following recommendations in order to implement successfully these technologies in a timely and cost-effective manner.

Recommendation #1. Comprehensive monitoring of mine drainage discharges. A concerted mine drainage discharge monitoring effort must be established to determine the current quality, quantity, spatial extent and temporal variability of all groundwater seepage points, boreholes, and other discharges in the mining area for at least one water-year. The Subcommittee realizes that successful treatment of the flows of entire surface waterways (Tar, Lytle and Beaver Creeks) is not feasible due to significant flow variations during extreme events, design considerations and lack of available land areas. Therefore, implementation of passive treatment systems must be targeted at specific discharges. Collection of the recommended data will assist in development of a Water Quality Improvement Prioritization Plan and the eventual realization of successful treatment throughout the mining impacted area. A project of this nature is scheduled to begin this Fall 2000 in the Beaver Creek watershed. This work funded by EPA will be performed by the University of Oklahoma School of Civil Engineering and Environmental Science, in cooperation with the Quapaw Tribe, State of Oklahoma and the U.S. Army Corps of Engineers, Tulsa District. Estimated costs for the full-scale effort are $1M.

Recommendation #2. Evaluation of water quality concerns from chat pile and millpond runoff. The contributions to surface water quality degradation from several major mine drainage discharge points have been well established, especially those near Mayer Ranch and Douthat Bridge. However, the contribution of chat pile and millpond runoff has not been determined. The Subcommittee therefore recommends a similar water quality and quantity monitoring effort be conducted at selected areas to determine the contribution of runoff waters to overall degradation of surface water bodies. These monitoring efforts, both at groundwater discharge and surface runoff points should begin as soon as possible, preferably in Fall 2000. Estimated costs are $1M.

Recommendation #3. Implementation of pilot-scale demonstration systems. Once the recommended data collection effort has been completed, the Subcommittee recommends the implementation of pilot-scale demonstration passive treatment wetland systems be conducted at representative discharges. The subsequent evaluation of performance of these systems will provide valuable information for the design, sizing and construction of future full-scale treatment systems. The overall applicability, sequencing and design of the coupled aeration and sulfate reduction systems could then be evaluated and adjusted prior to full-scale implementation throughout the watershed. Sites for demonstration projects have been identified at the Mayer Ranch discharges near Commerce and in the Beaver Creek watershed. It is anticipated that similar passive treatment demonstration systems should be constructed at selected chat pile and millpond runoff areas. The feasibility study for the Mayer Ranch pilot scale demonstration funded by U.S. Army Corps of Engineers should be complete and implementation occur by Fall 2001. The feasibility study for the Beaver Creek pilot scale demonstration funded by EPA should also be completed by Fall 2001. Estimated costs are $5M.

Recommendation #4. Coordination with other subcommittees and the U.S. Army Corps of Engineers Wetland Concept Plan. Many of the water quality issues at the Tar Creek site may be impacted by the recommendations of various other Task Force subcommittees, especially Drainage and Flooding, Subsidence, Chat Use, and Native American Issues. In addition, the U.S. Army Corps of Engineers Wetland Concept Plan incorporates passive treatment wetlands as a design component. The Subcommittee recognizes that it is imperative that these efforts be fully and completely coordinated, especially with regard to specific on the ground implementation efforts. These efforts should continue in Fall 2000. Estimated costs are $100K.

Recommendation #5. Full-scale treatment system implementation. The water quality concerns at the Tar Creek Superfund site are neither irreversible nor intractable. Collection of adequate water quality and quantity data, performance information from pilot-scale demonstration projects and coordination with other remediation efforts will provide a successful recipe for full-scale implementation of passive wetland treatment systems. It is anticipated full-scale implementation could begin as early as Fall 2005. Estimated cost are $18M.

Literature Cited

Crisp, T., R.W. Nairn, K.A. Strevett, J.W. Everett and B. Chen. 1998. Passive treatment using coal combustion products: results of a column study. Abstracts, 25th Anniversary and 15th Annual National Meeting of American Society for Surface Mining and Reclamation, St. Louis, MO.

EPA. 1994. Five Year Review, Tar Creek Superfund Site, Ottawa County, Oklahoma.

Hedin, R.S. and R.W. Nairn. 1993. Contaminant removal capabilities of wetlands constructed to treat coal mine drainage. *In*: G.A. Moshiri (ed.), Constructed Wetlands for Water Quality Improvement, CRC Press, Lewis Publishers, Inc. Chelsea, MI pp. 187-195.

Hedin, R.S., R.W. Nairn and R.L.P. Kleinmann. 1994. Passive Treatment of Coal Mine Drainage. U.S. Bureau of Mines Information Circular 9389. 37 pp.

Jage, C.R., C.E. Zipper and A.C. Hendricks. 2000. Factors effecting performance of successive alkalinity producing systems.. 2000. In: W.L. Daniels and S.G. Richardson (eds.), A New Era of Land Reclamation, Proceedings, 2000 Annual Meeting of the American Society for Surface Mining and Reclamation, Tampa, FL, pp. 451-458.

Kepler, D.A. and E.C. McCleary. 1994. Successive alkalinity producing systems (SAPS) for the treatment of acidic mine drainage. U.S. Bureau of Mines Special Publication SP 06A-94, Volume 1. pp. 195-204.

Nairn, R.W. and R.S. Hedin. 1992. Designing wetlands for the treatment of polluted coal mine drainage. *In*: M. C. Landin (ed.), Wetlands: Proceedings of the 13th Annual Conference of the Society of Wetland Scientists, New Orleans, LA, South Central Chapter, Society of Wetland Scientists, Utica, MS, pp. 224-229.

Nairn, R.W., M.N. Mercer, and J.W. Everett. 1999. Passive treatment using coal combustion products: An innovative vertical flow constructed wetland field study. *In*: Mining and Reclamation for the Next Millenium, Proceedings, 16th Annual National Meeting, American Society for Surface Mining and Reclamation, Scottsdale, AZ.

Nairn, R.W., M.N. Mercer and S.A. Lipe. 2000. Alkalinity generation and metals retention in vertical flow treatment wetlands. In: W.L. Daniels and S.G. Richardson (eds.), A New Era of Land Reclamation, Proceedings, 2000 Annual Meeting of the American Society for Surface Mining and Reclamation, Tampa, FL, pp. 412-420.

Nuttall, C. 1999. Cleaning zinc out of the River Nent. Geoscientist 9:4-6.

Nuttall, C.A. and P.L. Younger. 2000. Zinc removal from hard, circum-neutral mine waters using a novel closed-bed limestone reactor. In press in Water Research

Oklahoma Water Resources Board. 1991. Tar Creek After Action Monitoring Report. Xx pp.

Peart, D. and C. Cooper. 2000. Preliminary investigation of influent distribution in a vertical flow system. In: W.L. Daniels and S.G. Richardson (eds.), A New Era of Land Reclamation, Proceedings, 2000 Annual Meeting of the American Society for Surface Mining and Reclamation, Tampa, FL, pp. 427-437.

University of Oklahoma Environmental Capstone Class. 2000. Development of Holistic Remediation Alternatives for the Catholic 40 and Beaver Creek. Report submitted to Governor Frank Keating's Tar Creek Superfund Task Force,April 28, 2000. 45 pp. + appendices.

Watzlaf, G.R., K.T. Schroeder and C. Kairies. 2000. Long-term performance of alkalinity-producing passive systems for the treatment of mine drainage. In: W.L. Daniels and S.G. Richardson (eds.), A New Era of Land Reclamation, Proceedings, 2000 Annual Meeting of the American Society for Surface Mining and Reclamation, Tampa, FL, pp. 262-274..

Willow, M., R.R.H. Cohen, and L. Thompson. 1998. Factors affecting the efficiency of passive mine drainage treatment systems. Tailings and Mine Waste 98: 899-907.

Table 1. Applicable mine drainage treatment technologies for net alkaline Tar Creek discharges where neutralization is not necessary.

Process	Effect	Implementation Type	Technical attributes	Estimated costs
Reverse osmosis		Chemomechanical assembly	Complex yet established; high & continuous energy operation & maintenance costs; size unknown; potential for fouling.	Capital: $120,000,000 O&M: $100,000/yr
Ion exchange	Substitution of less desirable chemicals with salts	Reaction vessels filled with exchange media	Complex yet established; high rates of media consumption & regeneration; high operation & maintenance costs; potential for fouling	Capital: $70,000,000 O&M: unknown
Electrodialysis		Electromechanical assembly	Complex technique; high energy and operation & maintenance costs; size unknown; potential for fouling	Capital: $50,000,000 O&M: unknown
Aeration	Iron oxidation, hydrolysis and precipitation; limited toxic metal removal	Active aeration, passive oxidation ponds, natural wetlands	Simple, well-established non-biological process; little to modest external energy costs; very reliable; low operation & maintenance; potentially large area required	Capital: $350,000 O&M: $10,000/yr
Sulfate reduction	Precipitation of toxic metals as insoluble sulfides or carbonates; production of additional alkalinity	Passive anaerobic wetlands; constructed bioreactors; natural wetlands	Established biological and geochemical processes; based on naturally occurring constituents already in water; limited external energy; low operation & maintenance; potentially large area required	Capital: $15,000,000 O&M: included in aeration costs

Table 2 Active Treatment Technologies

Name	Description	Materials	Places Used	Results	Cost
Aeration/ Oxidation	The process of introducing air into water. Oxidation occurs when oxygen in air combines with metals in the water. If the water is oxidized, metals generally will precipitate at lower pH values. However, only about 10 mg/L O2 can dissolve in water, thereby limiting the oxidizing effects of water not directly exposed to air.	AMD H2O Lagoon/treatment cell Aeration units		Causes iron to precipitate	Capital Cost: Aeration (mechanical) 0.05 mgd (0.08 cu.ft./sec) $35,000 150 mgd (232 cu.ft./sec) $1,980,000 Aeration (diffused) 0.05 mgd (0.08 cu.ft/sec) $40,000 150 mgd (232 cu.ft/sec) $1,386,000
Calcium Carbonate	Limestone has been used for decades to raise pH and precipitate metals in AMD. It has the lowest material cost and is the safest, easiest to handle of the AMD chemicals, and produces the most compact and easy to handle sludge material. Low solubility, especially in cold weather, its tendency to develop an external coating, or amor, of ferric hydroxide when added to AMD, and its inability to raise pH to sufficient levels for Mn removal. Limestone has also been used to treat AMD in anaerobic drains and aerobic environments.	CaCO3 Storage unit Dispensing unit	Rotary drum stations have been used to grind limestone into a powder before introduction into streams and have been constructed in West Virginia for treating AMD streams.	Limestone introduction into the river by rotary drums has restored 22 km (14 miles) of the Blackwater River below the drum station and maintained the pH above 6.0. A fish survey in 1995 showed 17 species, including rainbow and brown trout, inhabiting the river.	A six-month drum station was constructed in 1994 on the Blackwater River at a cost of $900,000, and it introduces about 90 grams/sec of ground limestone to the stream or about 8.6 Mg/day (9.5 tons/ day), at a dosage of 28 grams per cubic meter water flow (28 mg/L). The drum station uses about 1800 Mg (2,000 tons) of limestone in an average river flow year at a cost of $12.60/Mg ($14/ton) of limestone delivered.
Trapzene	Trapzene (CaO2) is the trade name for a specially formulated compound of calcium peroxide. It is used as an oxidant as well		Lilly and Ziemkiewicz report successful treatment of Mn at several sites.	Water pH was raised from 3.5 to 7.5 with Trapzene application and metals (Fe, Mn, and Al) were removed	

	as an acid neutralizer. It seems to be especially useful for Mn oxidation and removal.			at a lower pH than had been achieved with liquid NaOH. Sludge volumes were also reduced using Trapzene compared to NaOH.
Calcium Hydroxide	Hydrated lime, $Ca(OH)_2$, is the most commonly-used chemical for treating AMD. It is sold as a powder that tends to be hydrophobic, and extensive mechanical mixing is required to disperse it in water. Hydrated lime is particularly useful and cost effective in large flow, high acidity situations where a lime treatment plant with a mixer/aerator is constructed to help aerate the water and mix the chemical with the water.		Central Ohio Coal Company switched from a 20% caustic solution to high calcium $Ca(OH)_2$. The raw water is treated with $Ca(OH)_2$, then aerated, and the solids are settled in a sedimentation basin.	The resulting lime sludge settles relatively quickly and the final settled volume is less than that of the caustic sludge. Final water quality is within effluent limits and can be discharged.
Caustic Soda	This method is often used in remote locations and in low flow, high acidity situations. It is commonly the chemical of choice if Mn concentrations in the AMD are high because caustic can raise water pH to 13.0. The major drawbacks of using liquid caustic for AMD treatment are high cost, dangers in handling the chemical, and high sludge volumes.		Southern Ohio Coal Company uses a 50% caustic solution to treat AMD that is eventually recycled to its preparation plant after solids are settled. Concerns about the use of lime and gypsum precipitation in the return water makes caustic treatment the preferred choice over lime treatment. The raw water is treated with 50% liquid NaOH, then	After flocculant addition, the water and solids enter a large thickener designed to receive the high flow of water. Approximately 10% of the water treated is removed as sludge. Final water quality meets effluent limits.

		aerated by floating aerators, and a flocculant is added to aid in solids settlement.	
Ammonia	Ammonia dissolves readily when released into water. It behaves as a strong base and can easily raise the pH of receiving water to 9.2. Injection of ammonia into AMD is one of the quickest ways to raise water pH. It should be injected into flowing water at the entrance of the pond to ensure good mixing because ammonia is lighter than water. A cost reduction of 50% to 70% can be realized when ammonia is substituted for caustic if the target pH for metal precipitation is less than 9.2.		Skousen found a 73% reduction in cost when switching from 20% NaOH to ammonia. This figure was based on a 950-L flow with an acidity concentration of 500 mg/L as $CaCO_3$. The annual cost to treat this drainage with ammonia was \$32,000 compared to \$121,000 with 20% NaOH.

Reverse Osmosis	Osmosis occurs if two solutions of different concentrations in a common solvent are separated from one another by a membrane. If the membrane is semi-permeable then the solvent will flow from the more dilute solution to the more concentrated solution until an equilibrium concentration is reached. In a reverse osmosis, the direction of solvent flow is reversed by applying pressure to the more concentrated solution. The process produces a high quality effluent water suitable directly for potable and industrial use. The concentrated brine solution is high in acid, iron, and sulfate.		Capital Cost: 0.05 mgd (0.08 cu.ft./sec) $290,000 150 mgd (232 cu.ft./sec) $11,500,000
Calcium Oxide	Pebble quicklime, CaO, has been recently used in conjunction with a water wheel application system. This system was initially used for small and/or periodic flows of high acidity because calcium oxide is very reactive. Recently, however, water wheels have been attached to large bins or silos for continuous treatment of high flow/high acidity	One operator in northern West Virginia calculated a water wheel unit paid for itself in just 105 days of operation at their site.	With the water wheel, the same water was treated at a cost of about $75 per day or about $28,000 per year. Three other sites showed between 62 and 82% cost savings when using quicklime vs caustic.

	situations.			
Ion Exchange Resins	Iron exchange in water treatment is defined as the reversible interchange of ions between a solid medium and the aqueous solution. The most common ion exchange example is the softening of “hard” water for domestic use. The hard water (caused by Ca2+ and Mg2+ ions in solution) is passed through a bed of ion exchange material, which is charged with monovalent cations, usually sodium.			Capital Cost: 0.05 mgd (0.08 cu.ft./sec) $46,000 150 mgd (232 cu.ft./sec) $6,300,000
Electrodialysis	This usually uses many thin compartments of solution separated by membranes that permit passage of either positive ions (cations) or negative ions (anions)and block passage of the oppositely charged ions. Cation-exchange membranes are alternatively stacked with anion-exchange placed between two electrodes. The solution to be treated is circulated through the compartments and a		Hilton found that electrodialysis worked well in ponds of acid mine drainage, but found the membranes to clog very quickly with metal ions. Iron quickly fouls the membranes and causes problems for disposal.	Capital Cost: 0.05 mgd (0.08 cu.ft./sec) $320,000 150 mgd (232 cu.ft./sec) $48,400,000

	direct current power source is applied. All cations gravitate toward the cathode (negatively terminal) and transfer through one membrane, while anions move in the opposite direction, thereby concentrating in alternative compartments.		
Metals Recovery from AMD Sludge	AMD treatment by chemicals causes the formation and precipitation of metal hydroxides in ponds. Passive treatment of AMD also accumulates metal hydroxide sludges into discrete area. This sludge contains various concentrations of metals corresponding to the amounts in the source water. Since most mine drainage contains some level of Fe, the possible recovery and utilization of Fe hydroxides, oxyhydroxides, or oxides as sources of Fe for pigments, coatings, catalysts, and foundry sands. Other metals, if sufficient in quantity in the source AMD, may also be recoverable for industrial and commercial uses.		Fish et al. Found Fe oxides from alkaline wetland sludges to be similar or slightly inferior in comparison to natural and synthetic Fe oxide products. Rao et al. Showed Fe and Zn could be recovered from AMD , but that a three-step process was needed for separation of metals.

Natural Zeolites	Natural eolites are hydrous aluminosilicates that may be used to exchange ions for treatment of AMD. The sodium ion, naturally occurring in zeolites, is preferentially exchanged for metal cations. Once the zeolites were loaded and filled with exchanged metal cations, the material must necessarily be regenerated using a sodium chloride solution to remove the metal cations from the aluminosilicate matrix.		The U.S. Bureau of Mines conducted several laboratory studies which demonstrated natural zeolites were successful in reducing metal concentrations in AMD to drinking water standards, but no field tests were initiated.

Table 3 Passive Systems Technologies

Name	Description	Materials	Places Used	Results
Constructed Wetlands: Aerobic	Man-made ecosystems that mimic their natural counterparts. Generally used to collect water and provide residence time and aeration so metals in the water can precipitate. Water usually has net alkalinity. Metals precipitate as they oxidize and precipitates are retained in the wetland or downstream. Plants encourage more uniform flow and effective wetland area for water contact.	Overland flow Cattails planted in substrate Wetland vegetation planted in shallow, relatively impermeable sediments comprised of soil, clay, or mine spoil.	G.A. Brodie and coworkers have reported extensively on their use of these wetlands. Nine wetlands received moderate quality AMD which required no further post-system treatment of water exiting the wetlands. Four wetlands treated water with high Fe and no net alkalinity. Two of these required NaOH treatment to comply with NPDES effluent limits, and two others used ALDs for further treatment of the effluent. A final wetland received low Fe and Mn and is ineffective in Mn removal.	Based on their experience with these systems since 1985, Brodie suggested that staged aerobic wetland systems could accommodate Fe loads of up to 21 grams/meters squared/day even in the absence of excess alkalinity. Manganese loads up to 2 grams/meters squared/day can be accommodated, if alkalinity is present.
Constructed Wetlands: Horizontal Flow Anaerobic	Encourage interaction of water with organic-rich substrates, which contribute significantly to treatment. The wetland substrate may contain a layer of limestone in the bottom of the wetland or the limestone may be mixed among the organic matter. Wetland plants are transplanted into the organic substrate. These systems are used when the water has net acidity, so alkalinity must be generated in the wetland and introduced to the net acid water in order to accomplish significant precipitation of dissolved metals.	Horizontal flow above organic substrate Wetland vegetation planted into deep, permeable sediment comprised of soil, peat moss, spent mushroom compost, sawdust, straw/manure, hay bales, or a variety of other organic mixes, which are often underlain or admixed with limestone.	A field study, which examined five wetland substrate types over a 25 month period, also demonstrated that organic substrates were saturated after only one to seven months of AMD input at 9 to 17 mg Fe per gram substrate.	Although some natural inputs of organic matter occur annually at plant senescence, the adsorption capacity of a wetland is limited by saturation of all exchange sites. Substantial artificial inputs of organic matter or fertilizer have been used as a successful strategy to temporarily renew this adsorption capacity, following an observed decline in performance.

Anoxic Limestone Drains	Buried cells or trenches of limestone into which anoxic water is introduced. The limestone dissolves in the mine water and adds alkalinity. The sole function is to convert net acidic mine water to net alkaline water by adding bicarbonate alkalinity. The removal of metals within an ALD is not intended and has the potential to significantly reduce the permeability of the drain resulting in premature failure.	Horizontal flow through buried limestone	These systems were used at the Howe Bridge and Morrison.	Alkalinity in effluents increased by 128 mg/L at Howe Bridge and 248 mg/L at Morrison over influent water. CO2 pressures were near 0.1 atm and calcite was at about 10% of saturation. For the past eight years, the effluent from the ALD-wetland system at Morrison has always met effluent criteria (pH 6-9 and Fe less than 3 mg/L. At Howe Bridge, the ALD-wetland system has removed an average of 70% of the Fe over the past seven years.
Successive alkalinity producing systems (SAPS)	In these vertical flow systems, water flows downward, usually from a pond, through organic matter and usually through limestone before flowing out of the system through a drainage system. These systems greatly increase the interaction of water with organic matter and limestone. In addition, passage through an organic layer removes oxygen and Fe+3, which are limitations for ALDs.	Vertical flow through an organic layer overlying a limestone bed		
Open Limestone Channels	Introduce alkalinity to acid water in open channels or ditches lined with limestone. Acid water is introduced to the channel and the AMD is treated by limestone dissolution. Armoring of the limestone with iron hydroxides reduces limestone dissolution, so longer channels and more limestone is required for water treatment.		At the Brandy Camp site in Pennsylvania, an open limestone channel was employed to treat AMD with a pH of 4.3, acidity of 162 mg/L as CaCO3, Fe of 60 mg/L, Mn of 10 mg/L, and Al of 5 mg/L.	After passage through the open lime channel, the effluent had a pH 4.8, net acidity of 50 mg/L as CaCO3, Fe of 17 mg/L, Mn of 8 mg/L, and Al of 3 mg/L. The channel removed 72% of the Fe and about 20% of the Mn and Al from the water.

Bioremediation	Bioremediation of soil and water involves the use of microorganisms to convert contaminants to less harmful species in order to remediate contaminated sites. Microorganisms can aid or accelerate metal oxidation reactions and cause metal hydroxide precipitation. Other organisms can promote metal reduction and aid in the formation and precipitation of metal sulfides. Reduction processes can raise pH, generate alkalinity, and remove metals from AMD solutions.		A mixture of organic materials (sawdust and sewage sludge) was emplaced into a mine spoil backfill to simulate microbial growth and generate an anoxic environment through sulfate reduction.	The results of the organic matter injection process caused no change in water pH, about a 20% decrease in acidity (1500 to 1160 mg/L as CaCO3) and a similar decrease in Fe, Mn, and Al. The results indicate that the process works, but improvements in organic material injection and the establishment of a reliable saturated zone in the backfill are needed for maximum development.
Diversion Wells	A simple device initially developed for treatment of stream acidity caused by acid rain. It has been adopted for AMD treatment. A typical diversion well consists of a cylinder or vertical tank of metal or concrete filled with sand sized limestone. The acid water dissolves limestone for alkalinity generation, and metal flocs produced by hydrolysis and neutralization reactions are flushed through the system by water flow out the top of the well. Limestone dissolution helps remove Fe oxide coatings so that fresh limestone surface are always exposed.		At the Galt site in West Virginia, a diversion well changes a 20 L/min flow from a pH of 3.1 to 5.5, acidity from 278 to 86 mg/L as CaCO3, Fe from 15 to 2 mg/L, and Al from 25 to 11 mg/L.	

Limestone Sand Treatment	Sand-sized limestone may also be directly dumped into AMD streams at various locations in watersheds. The sand is picked up by the stream flow and redistributed downstream, furnishing neutralization of acid as the stream moves the limestone through the streambed. The limestone in the streambed reacts with acid in the stream, causing neutralization.		The West Virginia Division of Environmental Protection treats 41 sites in the Middle Fork River, including the headwaters of 27 tributaries. The first year's full treatment was based on four times the annual load for non-AMD impacted streams and two times the load for AMD tributaries.	Water pH has been maintained above 6.0 for several miles downstream of the treatment sites. It is predicted that treating the river with limestone sand will be necessary three times a year to maintain water quality for fish populations.

Table 4 In-situ control Technologies

Name	Description	Materials	Places Used	Results	Cost
Bactericides	Anionic surfactants are used to control bacteria that catalyze the conversion of Fe^{2+} to Fe^{3+}, which thereby control pyritic oxidation. They are used primarily in situations where immediate control of AMD formation is important and work best on fresh, unoxidized surfaces. Bactericides are often liquid amendments, which can be applied to refuse conveyor belts or sprayed by trucks on cells of acid-producing materials in the backfill. Bactericides have also been used successfully at metal mines.		One example of surfactant application was done in 1988 on a 4.5 ha refuse site in Pennsylvania. Surfactant was applied via a hydroseeder at rates of 225 kg/ha initially, then successive amounts were added as fresh refuse was deposited.	Effluent from the pile showed a 79% decrease in acidity and an 82% decrease in Fe.	Cost savings at the AMD plant were $300,000 per year.
Alkaline Addition: Limestone	Limestone is often the least expensive and most readily available source of alkalinity. It has a Neutralization Potential (NP) of between 75 and 100% and is safe and easy to handle. On the other hand, it has no cementing properties and cannot be used as a barrier. Because of limestone's limited solubility, it can only raise the pH of a system to approximately 8.3. Under certain situations, primarily when Mn levels must be decreased, higher pH levels are required.		Field and laboratory studies have indicated that there are threshold levels of NP above which acid conditions in coal mines do not develop when the mines are properly reclaimed.	Bradham and Caruccio found that within the northeastern U.S. coal fields, at 3.7% NP no acid would be produced at a 95% confidence level.	

Alkaline Addition: Calcium and Magnesium Oxides	Materials that can increase water pH above 8.3 include: calcium-magnesium oxide (Magnalime), CaO, and Ca(OH)2.	Alkaline recharge trenches have also been constructed with CaO and Ca(OH)2 on top of an 8 ha coal refuse disposal site, which produced AMD seepage. After installing the alkaline recharge pools, acidity reductions of 25 to 90% were realized with concomitant 70 to 90% reductions in Fe and sulfate in seepage water.	The following conclusions and recommendations were made: (1) use highly soluble alkaline materials; (2) maximize water volumes through trenches by directing surface water flow into the pool; (3) use multiple alkaline recharge pools to increase chances of influencing groundwater flows; (4) construct infiltration paths into the backfill to improve alkaline diffusion and flushing; and (5) allow sufficient time for the effect to become apparent.
Alkaline Addition: Coal Combustion By-Products	Fluidized Bed Combustion (FBC) ash is produced at power generating plants that burn high sulfur coal or refuse in an FBC system Sulfur dioxide emissions are controlled by injecting limestone into the combustion bed. At combustion temperatures, the limestone calcines leaving calcium oxide. About one-half of the CaO reacts with sulfur dioxide to form gypsum and the rest remains unreacted. Therefore, FBC ashes generally have NP's of between 20 to 40% and they tend to harden into a cement after wetting. Other power generation ashes, like flue gas desulfurization (FGD) products and scrubber sludges, may also have a significant NP which make them suitable alkaline amendment materials.	Researchers from American Electric Power and Ohio State University used FGD material to amend coal refuse in field and laboratory studies at the Rehobeth site. The FGD was determined to have a neutralizing potential of 15% CaCO3 equivalency and a permeability of 1 x 10 –6 cm/sec.	The FGD material produced a larger yield of vegetation than refuse amended with agricultural lime at equivalent amounts. Water quality from the field experiments met drinking water drinking water standards, while untreated (control) runoff had high acidity and metal concentrations. Additionally, FGD material test plots had reduced infiltration rates, were highly stable, and resisted erosion.

Alkaline Addition: Kiln Dust and Steel Slags	Kiln dust, produced by lime and cement kilns, contains 15 to 30% CaO with the remaining 70 to 85% of the material being hydrated lime and limestone. Kiln dust absorbs moisture and also hardens upon wetting. It is widely used as a stabilization and barrier material. Steel making slags are locally available in large quantities at low cost and, when fresh, have NP's from 45 to 90%. Studies indicate that columns of steel slag maintain constant hydraulic conductivity over time and produce highly (greater than 1,000 mg/L as CaCO3) alkaline leachate. Steel slag can be used as an alkaline amendment as well as a medium for alkaline recharge trenches. Slags are produced by a number of processes so care is needed to ensure that candidate slags will not leach metal ions such as Cr, Mn, Ni, or Pb.	Lime kiln dust has also been added to the overburden of a coal mine having alkaline deficiencies of up to 2450 Mg CaCO3/ha (1090 tons CaCO3/ac). The dust was applied to the pit floor at about 224 Mg/ha (100 tons/ac), also around special-handled cells, and also on fractured overburden after blasting. After grading, the dust was applied to the backfill surface, back-dragged by bulldozers to afford some mixing, and then topsoiled.	Compared to an acid discharge from an adjacent mine in the same coal bed, the water quality from this amended site is alkaline with low metal concentrations.
Alkaline Addition: AMD Sludge	Addition of alkaline sludges or flocs, generated from the neutralization of AMD, to the surfaces of backfills or to acid-producing materials during mining and backfilling may provide several benefits. First, flocs from all neutralization processes contain excess alkalinity and this alkalinity may be used to neutralize acid-producing materials or acidic water in the backfill. Lime-treated flocs often contain up to 50% unreacted lime, which may be used to further neutralize acidity. Second, disposal of the metal flocs from AMD treatment ponds is an expensive and long term problem and disposal in nearby surface mine soils or within	This technology was applied in the field at a coal refuse disposal area in West Virginia where dry AMD treatment sludge was used as a substitute for soil cover. The sludge was concentrated by pumping it to steel dumpsters lined with filter fabric, thereby allowing the water to filter into a sediment pool. This concentrated sludge was hauled to drying cells and then moved to the refuse pile for spreading and seeding.	The sludge generated by AMD neutralization with calcium oxide was well suited for establishing and supporting vegetation. Vegetation on this soil substitute surpassed growth on native soils. Water quality was positively influenced by the improved vegetation and metals were stabilized. Underground disposal of this material into abandoned deep mines on this property was initiated in 1997 and careful monitoring

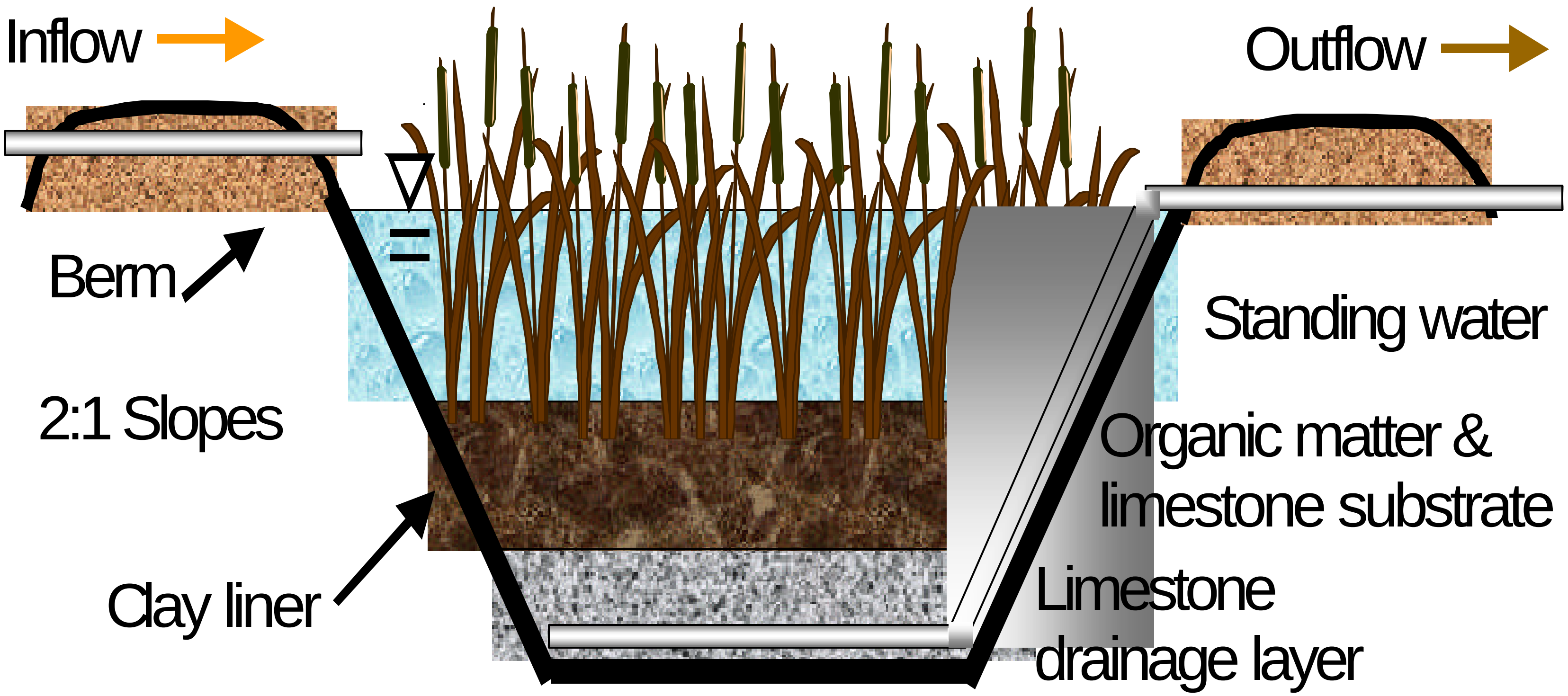
	spoil materials during mining can provide a cost effective alternative to other disposal options.			of all deep mine openings related to these underground works is being conducted.
Sewage Sludge	There is evidence from laboratory studies that the oxidation of pyrite can be inhibited by organic waste materials such as manures and sewage sludge. The inhibition process may be a combination of several mechanisms. First, Thiobacillus bacteria, which can catalyze iron oxidation, may convert from a chemoli- thotrophic bacteria to a heterotroph in the presence of readily decomposable organic matter. A second mechanism of inhibition may allow the sludge to complex Fe and eliminate it from oxidizing more pyrite, or adsorb/complex Al and other metal ions, thereby reducing hydrolysis and pH decreases. Sludge may also coat pyrite surfaces minimizing reaction surfaces. Decomposition of sludge consumes oxygen, thereby decreasing its availability for pyrite oxidation.		In 1977, digested and dewatered municipal biosolids were applied to an abandoned surface mine at the rate of 184 Mg/ha (85 tons/ac) on a 0.4 ha plot in Pennsylvania. Data were collected for a 5 year period and the site was re-evaluated after 12 years.	The results showed that vegetation could be established and that the groundwater quality improved. The State of Pennsylvania has used this technology to revegetate over 2,000 ha (5,000 ac) of mine land.

Encapsulation	This process encircles or covers acid-producing material with an impermeable material to limit its exposure to air and water. The material can be a synthetic liner or it may be a clayey material or other compacted material, which results in a layer with a low hydraulic conductivity. The material surrounding the acid-producing material may also be composed of alkaline material. Encapsulation can occur during the mining process where acid-producing materials are placed “high and dry” off the pit floor and away from the highwall. After placement and compaction (and treatment with alkaline material if required), the acid-producing material is then covered with local clayey material or topsoil and this material is also compacted on top of the acid- producing material. Backfilling then continues over the layers of acid and cover material until the final grade is achieved.	The plastic liner technology was applied at a surface coal mine in Alton, West Virginia. Approximately 18 ha (45 ac) of a combined refuse disposal and overburden site were covered with a 20 mil continuously-seamed PVC liner. The liner was placed over the surface of the backfill material and covered with 0.5 m (18 in.) of soil. Due to the steep slopes on the periphery of the backfill, approximately 3 ha (or about 20% of the site) were not covered.	The seeps and groundwater fluctuations from a larger, unlined portion of the mine were compared to seeps and groundwater levels from the PVC-lined area. The ground water wells in the lined area reflected a diminution in groundwater recharge. During the year following the installation, the seeps emanating from the PVC- lined area showed a 63% reduction in acid loads while there was no reduction in the seeps from the unlined portion.
Reclamation	Backfilling and revegetation together is one method of reducing acid loads from current mining operations or abandoned mine sites. Covering pyritic refuse or other acid-producing materials on a site with good soil material and establishing vegetation has a major impact on reducing acid concentrations in water and often decreases the flow of water from these sites by encouraging infiltration into soil and evapotranspiration by plants. If the majority of the water from an abandoned site is coming from	Water was reduced on 12 out of the 16 sites in West Virginia where reclamation was accomplished on bond forfeited and unreclaimed areas.	On those sites where flow was not reduced, water quality changed from acid to alkaline. In only two out of 16 cases was the acidity increased in the water, but flows were reduced dramatically causing a 45% decrease in total acid load. Vegetation establishment greatly reduced the occurrence and amount of runoff compared to a barren tailings area in

	underground mines, then surface treatments may have limited effects on reducing acid loads.		Montana. Runoff water from the vegetated area had a higher pH (6.2 vs 4.0), and metal loadings of As, Cu, and Zn were also more than four orders of magnitude lower than the unvegetated area.

Vertical-flow substrate wetlands

- Applicable to net acidic mine drainage
- Designed for alkalinity generation and/or metal sulfide formation



Aerobic wetlands

- ONLY applicable to net alkaline mine drainage
- Designed for metal oxidation, hydrolysis and precipitation